WATER QUALITY SIMULATION FOR PLANNING RESTORATION OF A MINED WATERSHED

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Abstract. Water quality simulation was performed to evaluate the effects of restoration alternatives on metals transport in a mountainous watershed in Montana, U.S.A. impacted by hundreds of abandoned hardrock mines. The Water Quality Analysis Simulation Program (WASP5), developed by the U.S. Environmental Protection Agency (U.S. EPA), was used to assist in planning restoration of the Upper Tenmile Creek watershed, a major drinking water supply for the City of Helena. Synoptic survey data collected by U.S. EPA and the U.S. Geological Survey were used for model calibration and validation. The effectiveness of eight restoration alternatives was modeled under steady-state, low flow conditions. These alternatives ranged from removal of adit and point source discharges to modification of the water supply scheme to provide higher in-stream flows. The model was also used for a number of related purposes, including evaluation of metals loadings and losses, exceedances of water quality standards, interactions between metals in water and bed sediment, and model and data uncertainties. Although standards exceedances are common throughout the watershed, modeling results indicated that removal of point sources, mine waste near watercourses, and streambed sediment can help improve water quality. Alteration of the water supply scheme and increasing baseflow will also ultimately be required to meet standards for all metals. The model also showed that although adits and point sources contribute significant metals loadings to the stream during baseflow, in some areas shallow groundwater and bed sediment can also be sources of metals. Adsorption and precipitation onto bed sediments are also important loss mechanisms in some locations. The model helped to identify uncertainties in the metal partition coefficients associated with sediment, significance of precipitation reactions, and locations of unidentified sources and losses of metals.

Keywords: metals, mining, modeling, restoration, simulation, water quality, WASP, watershed

1. Introduction

Impairment of stream water quality due to increased loadings of metals to aquatic systems from hardrock metal mine waste is a common problem in high elevation watersheds in the western United States and throughout the world. Impacts to aquatic ecosystems and biota are particularly widespread and severe due to elevated dissolved and particulate metals concentrations; enhanced erosion, transport and deposition of exposed sediment, tailings and waste rock; and low pH (U.S. EPA, 1997; Caruso and Ward, 1998; Gurrieri, 1998; Malmqvist and Hoffsten, 1999). Restoration of these impacted ecosystems is often complicated by the significant spatial and temporal variability in hydrology and steam chemistry (Sullivan

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and Drever, 2001), and geochemical reactions and attenuation of metals in the hyporheic zone (Fuller and Judson, 2000).

Simulation of water quality in receiving streams is often needed in these mining-impacted watersheds as an integral part of restoration planning and design to predict and evaluate the feasibility and effectiveness of potentially applicable restoration alternatives using quantitative information (Nimick and Moore, 1991; Simons *et al.*, 1995). Modeling metals fate and transport can greatly assist in understanding contaminant movement and other processes in the watershed and receiving stream as well as impacts on water quality and aquatic biota. Modeling can also aid in evaluating uncertainties, data gaps, and ultimately the effectiveness and sustainability of restoration efforts (Jenne and Zachara, 1987; Combest, 1991). Few studies, however, have been performed on the application of water quality simulation to evaluate the effectiveness of restoration strategies and the usefulness of this modeling in the restoration planning process in mining-impacted mountain watersheds. In the U.S., this modeling is needed for assessment and restoration of both abandoned and operational mines as well as for developing and implementing total maximum daily loads (TMDLs) and water quality restoration plans under the U.S. Clean Water Act of 1972. Similar needs exist for modeling metals transport in mining-impacted watersheds in other parts of the world.

There are few models available for evaluation of metals transport and restoration alternatives, and the applicability of existing models depends on the goals, scope, and sophistication of the modeling needed (Sanden, 1991; Bouchard *et al.*, 1995; Whitehead and Jeffrey, 1995; Banwart and Malmstrom, 2001; Runkel and Kimball, 2002). Models such as the U.S. Environmental Protection Agency's (U.S. EPA) Better Assessment Science Integrating Point and Nonpoint Sources (BASINS), Hydrologic Simulation Program Fortran (HSPF), and Enhanced Stream Water Quality Model (QUAL2E) are used widely for watershed and receiving water quality evaluation, but are not capable of modeling metals. Therefore, some studies have focused on equilibrium metals speciation models based on detailed pH and geochemical information, including MINTEQ (Pitt *et al.*, 1998), WATEQ (Williams and Smith, 2000) and PHREEQC (Tonkin *et al.*, 2002). Others have focused on estimating long-term contamination source strength, longevity, and possible future changes in discharge quality (Banwart and Malmstrom, 2000), or used a stochastic approach for concentrations or flows based on empirical data (Whitehead and Jeffrey, 1995; Caruso and Wangerud, 2002). These methods, however, do not directly model metals transport in streams, particularly as needed for restoration planning. Other studies have attempted to combine some of these models or geochemical information with simple hydrologic/flow models (Sanden, 1991; Broshears, 1996). These models, however, are generally not widely available for use in the majority of impacted watersheds. One of the few available models capable of modeling steady-state or dynamic metals transport in receiving streams is the U.S. EPA Water Quality Analysis Simulation Program (WASP5).

The Upper Tenmile Creek Watershed in Montana, U.S.A. has been severely impacted by hundreds of abandoned metal mines, is a major drinking water supply for the City of Helena, and requires the use of water quality simulation of metals transport to assist in restoration planning. The entire watershed has been designated as a Superfund site, and development of TMDLs is also required due to impairment of beneficial uses and water quality. Constituents of concern in surface water include arsenic (As), cadmium (Cd), copper (Cu), lead (Pb), and zinc (Zn) (CDM, 2001a). The objective of this study was to simulate the effects of potentially applicable watershed restoration alternatives on water quality and metals fate and transport in the Upper Tenmile Creek mainstem under low-flow, steady-state conditions using WASP5. This included evaluating the effectiveness of these alternatives in achieving restoration objectives and water quality standards. This modeling also allowed for the assessment of metals loadings and associated downstream water quality impacts; understanding of watershed and in-stream transport and fate mechanisms; and uncertainties associated with watershed processes, the model, and input/monitoring data.

2. Site Description

The Upper Tenmile Creek Watershed is 51 km^2, ranges in elevation from 1335 to 2485 m at the Continental Divide, and is located primarily in Lewis and Clark County, Montana (Figure 1). The watershed has a continental climate modified by Pacific Ocean air masses with cold, moist winters and warm, dry summers. The average annual precipitation measured from the nearest weather station (Frohner Meadows) is 624.8 mm (24.6 inches). The watershed is in the Northern Rocky Mountain physiographic province characterized by mountainous terrain with high and sharp relief formed from glaciation. Glacial features cover more than half of the basin and include cirque basins and moraine deposits, terrace and floodplains. Bedrock consists primarily of large areas of Cretaceous and Tertiary igneous rocks and small areas of Cretaceous sedimentary rocks. Mineralization during two periods resulted in silver-lead ore bodies, rich in galena and pyrite (FeS_2), and disseminated gold ore. Mountain slopes and ridges are part of unglaciated terrain, with a thin veneer of glacial deposits and alluvium in valleys (CDM, 2001a). Soils are typically well-draining sandy loams. Vegetation consists primarily of Ponderosa Pine and Douglas Fir forest, with some tundra grassland at the highest exposed elevations and willows, grasses, and sedges in lower riparian and wetland areas.

The mainstem of Upper Tenmile Creek is approximately 17 km long from the headwaters to the Helena water treatment plant, and the small town of Rimini is located approximately in the middle of the watershed adjacent to the creek. The watershed, like many other areas in this part of Montana, is rich in metal ore deposits associated with the pyrite. Gold, lead, zinc, and copper were mined between 1870 and the 1920s resulting in hundreds of abandoned mines and tailings areas.

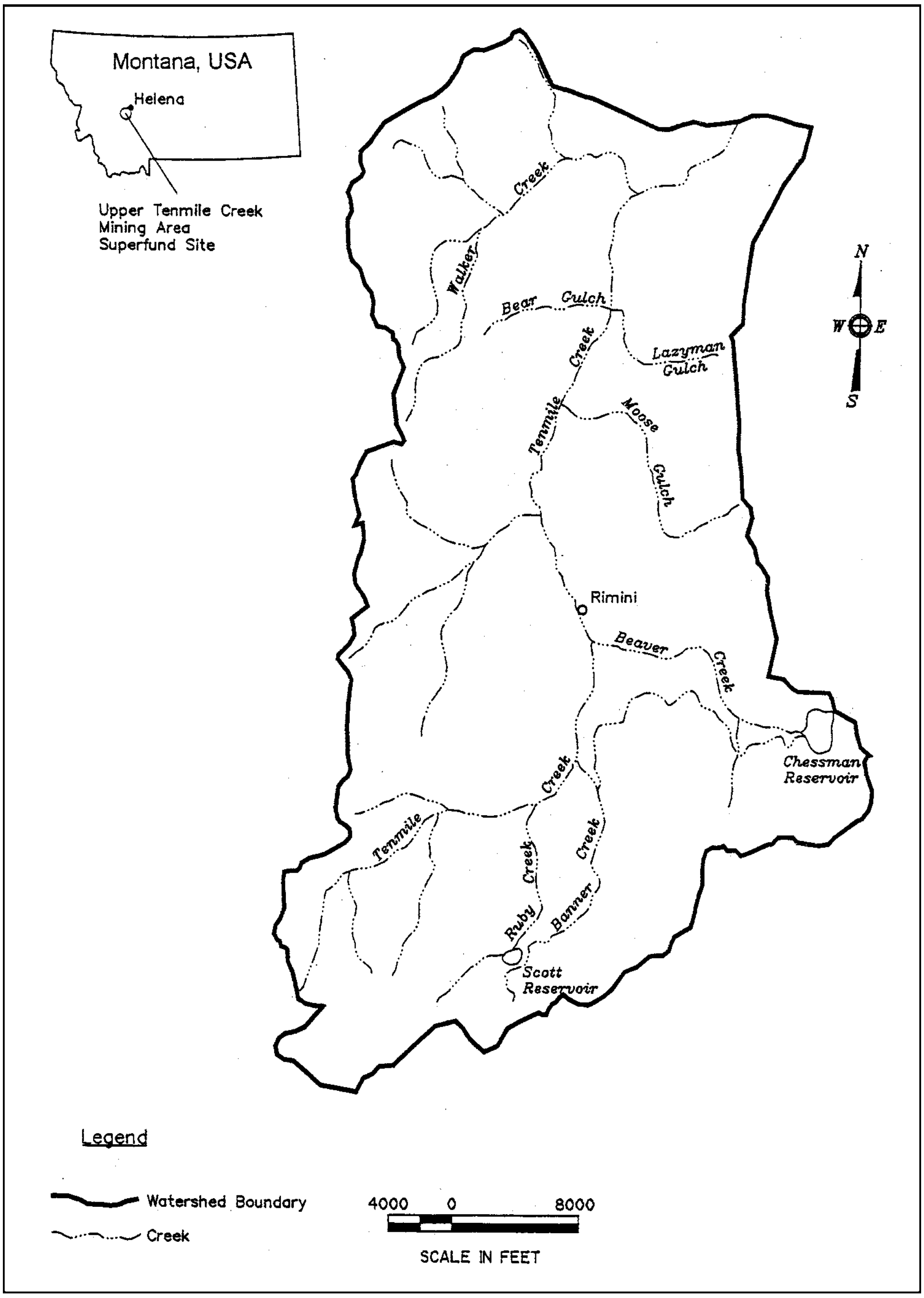

Figure 1. Location of Upper Tenmile Creek Watershed, Montana.

Significant loadings of metals, sediment, and acidity from these areas cause severe impairment of water quality and beneficial uses, particularly of aquatic ecosystems (CDM, 2001a).

3. Methods

WASP5 was used for modeling fate and transport of metals in the Upper Tenmile Creek Watershed and evaluating the effectiveness of a number of restoration alternatives. This model was developed by U.S. EPA to simulate surface water quality and 3-dimensional fate and transport of solutes in either the steady-state or dynamic mode (Ambrose *et al*., 1993). TOXI5 is the subcomponent toxics model used to estimate metals concentrations in water column or benthic/stream bed compartments throughout a water body at each time step in a simulation period, including interactions between the two types of compartments. Metals fate and transport processes simulated in WASP5 include loadings of point and nonpoint source water and constituents, including from tributaries, groundwater, and runoff; advection, dispersion and diffusion in stream segments; adsorption/desorption associated with sediment; precipitation/dissolution; and sediment transport and settling/scour of particulates (Figure 2).

U.S. EPA performed a synoptic water quality and flow sampling event in the watershed under base- or low-flow conditions in June 2000. The flow at the watershed outlet was approximately 0.1 $m^3\ s^{-1}$. Data from this survey were used for model calibration because it was the most complete data set available. The U.S. Geological Survey (USGS) also conducted a survey under higher flow conditions in June 1997 (flow of approximately 3 $m^3\ s^{-1}$). Data from this survey were used for model validation (see the Section Results and Discussion for details on model input, calibration, and validation). Significant increases in metals concentrations and loadings were observed in the mainstem immediately downstream of the Suzie Load and Lee Mountain Mine area during both sampling events.

Following calibration and validation, eight restoration alternative scenarios were modeled using steady-state, low-flow (June 2000 measured flows) conditions:

Alternative 1: Water treatment of adit discharges in the Rimini subarea achieving an 80% reduction in metals concentrations. These discharges include the Redwater, Suzie Load, and Lee Mountain adits.

Alternative 2: Water treatment of adit discharges in the Rimini subarea achieving reduction in metals concentrations to State of Montana water quality standards. These discharges also include the Redwater, Suzie Load, and Lee Mountain adits.

Alternative 3: Diversion, treatment, and consumption (no discharge back to Tenmile Creek) of adit discharges in the Rimini subarea through a community water system. These discharges also include the Redwater, Suzie Load, and Lee Mountain adits.

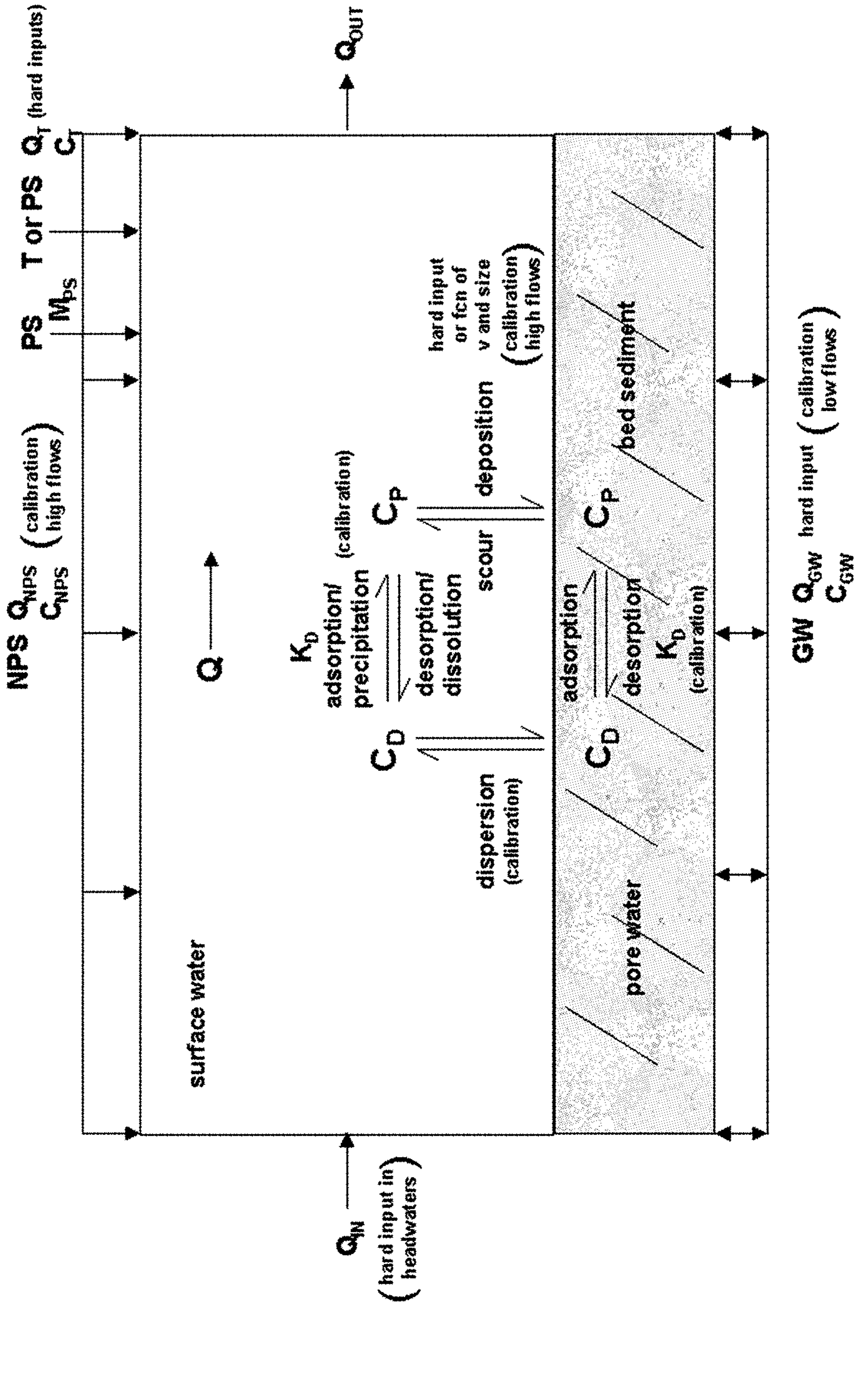

Figure 2. Schematic diagram of major metals fate and transport processes in WASP.

Alternative 4: Bypass 0.028 cubic meters per second (cms; 1 cubic foot per second (cfs)) of water through the City of Helena's Rimini diversion into Tenmile Creek.
Alternative 5(a): Combination of alternatives 1 and 4.
Alternative 5(b): Combination of alternatives 2 and 4.
Alternative 5(c): Combination of alternatives 3 and 4.
Alternative 6: Bypass 0.084 cm (3 cfs) of water through the City of Helena's Rimini diversion into Tenmile Creek.

Model inputs were those used for calibration, except where specific inputs were changed based on each restoration alternative. Modeled and measured total and dissolved concentrations and State of Montana water quality criteria for each metal were compared using graphs of concentration vs. distance from the headwaters of the main stem. The State criteria are for total recoverable metals, and the acute and chronic criteria vary along the length of the creek based on the hardness measured during June 2000.

Several methods were also used to evaluate the effects of existing bed sediment metals concentrations on the water column in Tenmile Creek, assuming that there were no other inputs of metals into the mainstem (that all other sources were remediated). One method involved using the WASP5 model, and two methods involved using methods outside of the model for this evaluation:

Method 1. The first method involved using the model with all metals loadings to the creek equal to zero, existing bed metals concentrations measured during the June 2000 synoptic survey, and the average equilibrium partition coefficient for dissolved and particulate forms of each metal estimated from laboratory desorption batch test results. The results represented equilibrium conditions in the water column with sediment as the only source of metals.
Method 2. The second method involved using the resulting equilibrium benthic porewater concentrations for each analyte based on the desorption batch test results. Results for four key locations were used: benthic segment 61 (near Rimini), benthic segment 62 (downstream from Moore Creek), Tenmile Creek below Suzie Adit, and Poison Creek near Red Mountain. Equilibrium low-flow conditions were assumed, where the mass in the porewater resulting from desorption from the bed sediment was distributed throughout both the porewater and overlying water column. Additional assumptions were that the bed sediment porosity = 0.25, bed sediment density = 2.5 kg L^{-1}, bed sediment depth = 0.3 m (1 ft), and water depth = 0.3 m (1 ft) under steady-state, low-flow conditions.
Method 3. The third method used a probablistic approach to evaluate the time required to scour and flush contaminated bed sediments from Tenmile Creek downstream (past the water treatment plant) based on high flow/storm/flood events. A preliminary hydrologic analysis of flows in the creek was performed, and high flows with associated return periods (probabilities) were estimated based on analysis of historic flows at the USGS gauging station in Tenmile Creek near Rimini (station 06062500).

Flows evaluated included those with a 1-, 5-, and 10-yr return period. These flows have probabilities of occurring in any given year of greater than 95, 20, and 10%, respectively. The estimated flows for these return periods are 286, 364, and 511 cfs, respectively. The 1-yr flow is also known as the mean annual flood (MAF).

Graphs developed by Colby (ASCE, 1975) based on empirical results from a range of rivers were used to estimate sediment discharge for the evaluation of sand (size range of 0.062 to 2 mm) at each site. This method is based on water velocity, median particle size (d_{50}) of sand, depth of flow, and water temperature. The d_{50} used for sand was 0.41 mm. Equations are currently not available for estimation of silt and clay discharge. Velocities for each selected location were calculated using the estimated flood discharge and the stream cross-sectional area. Parameters for this calculation were derived from existing data and hydraulic analysis using Manning's Equation (Chow, 1959) and the following assumptions and input parameters for dimensions of the channel and contaminated sediment:

Trapezoidal channel with 1:1 side slopes and 1.2 m (4 ft) bottom width;
Stream slope (S) = 0.0238;
Roughness coefficient (n) = 0.05;
Contaminated bed sediment depth = 0.3 m (1 ft), width = 1.22 m (4 ft), and length = 1.61 km;
(1 mile) (total volume of 595 m^3);
Bed sediment density = 2.5 kg L^{-1}.

The graphs developed by Colby were used in conjunction with the estimated velocity and assumed depth to compute the sand sediment discharge for each flood flow. The probability of removing a specific volume of sediment in any given year, and the total amount of time required to remove all of the assumed contaminated sediment, were then calculated. Contaminated bed sediment is predominantly fine-grained, i.e., sand-size particles (methods are currently not available for evaluation of discharge of silt and clay-size particles).

These analyses do not explicitly take into account the coarse-grained material and armoring of sands with gravels or cobbles in the creek that could reduce scouring and flushing of the finer-grained material. Therefore, the estimates presented here are probably conservative in that they might overestimate the quantity of contaminated sediment flushed and underestimate the time required for such flushing. They do, however, provide rough estimates of the quantities and timing of flushing.

4. Results and Discussion

The WASP5 model was constructed for steady-state, low-flow conditions in the Tenmile Creek mainstem based on constant metals loadings from tributaries, point

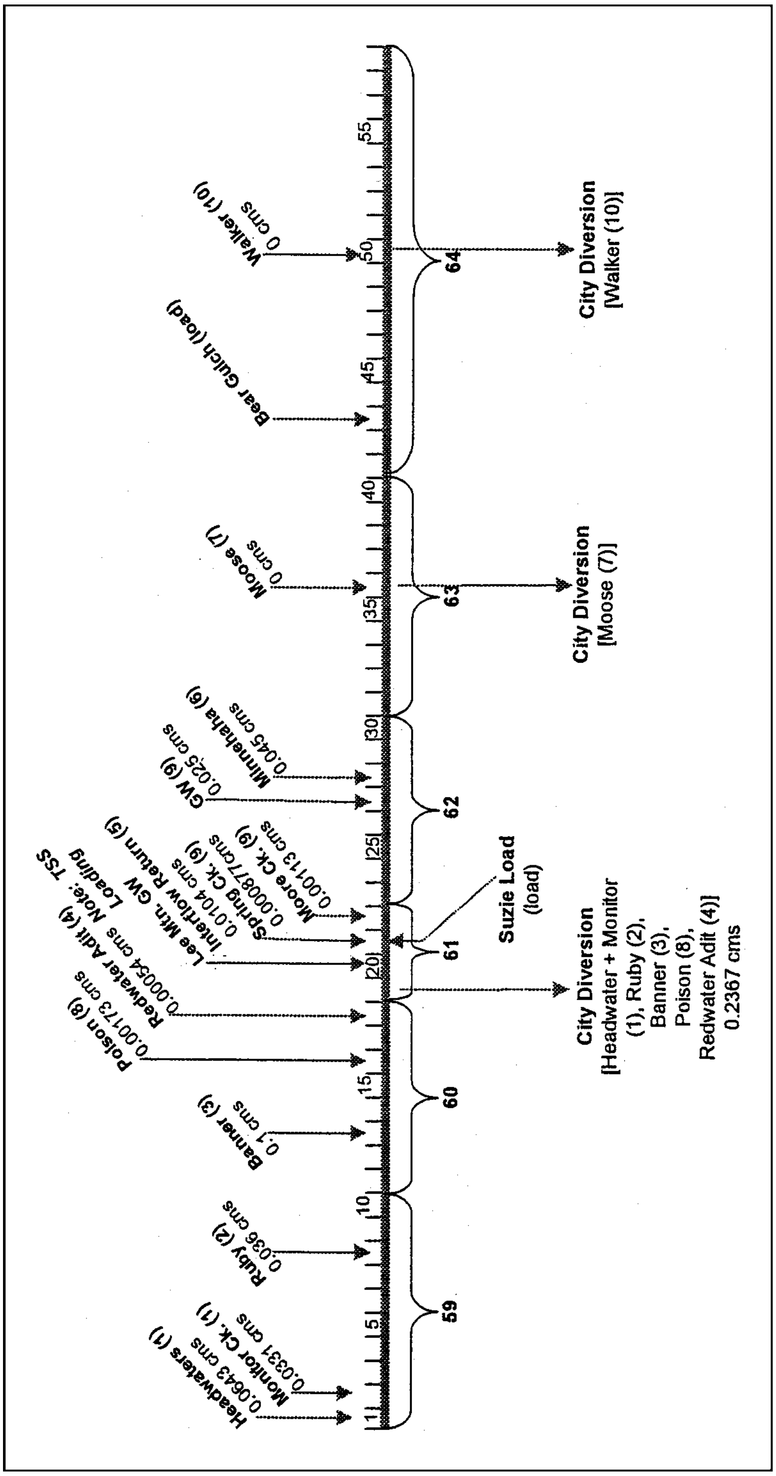

Figure 3. Schematic diagram for Upper Tenmile Creek mainstem WASP model with segmentation and flow inputs/outputs.

sources such as adits, and nonpoint sources. The model was used to calculate equilibrium concentrations and loadings of As, Cd, Cu, Pb, and Zn, extending over a length of approximately 17 km to the Helena water treatment facility (Figure 3). Fifty-eight uniform segments (length of approximately 300 m each) were used for the water column layer, and six segments were used for the underlying benthic layer with lengths varying from approximately 1200 to 5500 m. The length of each benthic segment was dependent on how many water column segments overly it.

Based on data from the two synoptic sampling events, user-specified headwater and tributary constant inflows were used to compute flows for each mainstem stream segment. The flows in the City of Helena water supply diversions were also included (Figure 3). The model also incorporated an apparent groundwater discharge to the stream near the Lee Mountain Mine that was observed during the June 2000 synoptic survey. Flow and channel measurements (average depths, widths, and velocities) made at two stations during June 2000 were used to define channel hydraulic parameters for the power equations within the model used to compute flow hydraulics.

Sediment loadings to the stream segments were estimated outside of WASP5 based on the available data, and sediment transport through the stream was modeled based on flows and vertical net settling/scour rates estimated by the user. Although vertical sediment transport rates are generally not significant under low-flow, steady-state conditions, they are important for modeling particulate metals fate and transport during dynamic storm and snowmelt events.

User-specified metal loadings were input as tributary flows and concentrations or point source mass loads (Table I). Metals transport was modeled via advection, diffusion and dispersion in two dimensions. A longitudinal dispersion coefficient of 1 $m^2\ s^{-1}$ was initially used, and vertical dispersion/diffusion was used to model the transport of dissolved metals between the water and underlying benthic layers. Bed sediment metals concentrations measured during June 2000 were used as initial values (Table II). Diffusion/dispersion was modeled with exchange coefficients and mixing lengths specified by the user. A 'lumped' partition coefficient (K_d) was used to simulate the distribution of metals between the dissolved and particulate phases. The particulate phase includes both sorbed and precipitated metals. This methodology can provide reasonable results because the pH in Tenmile Creek only ranges between 5 and 8.9 (based on June 2000 data). In systems where the pH is lower or more variable, the lumped K_d does not provide results that are as useful, and separate more detailed geochemical modeling of precipitation reactions should be considered. Partition coefficients were input by the user as spatially-variable values. Particulate metals transport was simulated based on fluvial sediment transport. Sorption/desorption of metals between the water column and underlying bed sediment was estimated using a dispersion/diffusion coefficient for exchange and partitioning between the dissolved and particulate fractions within the bed layer.

Model calibration was performed using data from the June 2000 synoptic survey and parameters including K_d values, vertical dispersion/diffusion coefficients, and

TABLE I

Upper Tenmile Creek flow, concentration, and loading model calibration inputs from June 2000

Seg-ment	Name	Distance (m)	Flow (cms)	As (mg/L)	As (kg/day)	Cd (mg/L)	Cd (kg/day)	Cu (mg/L)	Cu (kg/day)	Pb (mg/L)	Pb (kg/day)	Zn (mg/L)	Zn (kg/day)
1	Headwaters	152.4	0.064	0.0021	0.0117	0.00006	0.00032	0.0016	0.0089	0.00097	0.00539	0.0085	0.0472
2	Monitor	457.2	0.033	0.0009	0.0026	0.00092	0.00263	0.0002	0.0006	0.00200	0.00572	0.0006	0.0017
8	Ruby	2286	0.036	0.0032	0.0100	0.00030	0.00093	0.0023	0.0072	0.00140	0.00435	0.0453	0.1409
12	Banner	3505.2	0.100	0.0011	0.0095	0.00004	0.00033	0.0022	0.0190	0.00051	0.00441	0.0162	0.1400
16	Poison	4724	0.002	0.0090	0.0013	0.03300	0.00485	0.3300	0.0485	0.08700	0.01278	4.4000	0.6463
18	Redwater Adit	5334	0.001	0.1230	0.0057	0.03150	0.00147	0.0067	0.0003	0.00058	0.00003	7.2700	0.3392
19	City Diversion	5638.8	–0.237										
20	Lee Mtn	5943.6	0.010	0.0130	0.0117	0.00220	0.00198	0.0070	0.0063	0.01220	0.01096	0.3440	0.3091
21	Spring Ck/Suzie[a]	6248.4	0.001	0.0110	0.2208	0.00100	0.00598	0.0091	0.0036	0.00025	0.00029	0.0850	0.7904
22	Moores Ck	6553.2	0.001	0.0580	0.0057	0.00500	0.00049	0.0070	0.0007	0.00700	0.00068	0.6900	0.0674
27	Groundwater	8077.2	0.025	0.0450	0.0972	0.00150	0.00324	0.0070	0.0151	0.00000	0.00000	0.2000	0.4320
28	Minnehaha	8382	0.045	0.0030	0.0117	0.00200	0.00778	0.0060	0.0233	0.00050	0.00194	0.2300	0.8942
36	Moose	10820.4	0.000	0.0008	0.0000	0.00050	0.00000	0.0005	0.0000	0.00010	0.00000	0.0044	0.0000
43	Bear Gulch[a]	12954			0.0001		0.00003		0.0000		0.00003		0.0003
50	Walker	15087.6	0.000	0.0012	0.0000	0.00050	0.00000	0.0016	0.0000	0.00011	0.00000	0.0045	0.0000
		Totals	**0.081**		**0.3880**		**0.03001**		**0.1335**		**0.04658**		**3.8086**

All data obtained from WinWASP input (*.wif files).
All values are reported above the analytical detection limits.
[a] Suzie Load and Bear Gulch are input as loadings only with no contributing flow due to model limitations with regard to number of flow inputs.

TABLE II

Upper Tenmile Creek initial WASP model benthic sediment metals concentrations (mg kg^{-1})

Segment #	As	Cd	Cu	Pb	Zn
59	1	0.45	3.8	16	40
60	68	1.2	11	90	132
61	2770	40	47	97	4047
62	123	3	18	90	325
63	132	3	33	75	299
64	87	3	21	49	355

unaccounted flows. Vertical sediment exchange (settling/scour) rates were assumed to be zero for the low-flow, steady-state conditions. Calibration of flows indicated an unaccounted inflow to the mainstem upstream of Minnehaha Creek, which is probably from groundwater discharge or a small tributary. This was subsequently included in the model as part of the calibration. K_d values from the WASP5 User's Manual (Ambrose *et al.*, 1993) were initially used. These values can be a function of solids concentrations according to the manual and other references (U.S. EPA, 1989, 1996). Therefore, the higher the solids concentration, the higher the K_d, although some investigations and reviews of this effect have not supported this relationship (McKinley and Jenne, 1991). A relationship discussed in U.S. EPA (1996) was used with a solids concentration equal to 4 mg L^{-1} for water and ranging from 100 000 to 500 000 mg L^{-1} for the benthic layer. Manipulation of K_d values and dispersion/diffusion coefficients within ranges in the literature (Ambrose *et al.*, 1993) and of metals concentrations in unaccounted groundwater inflow was used to calibrate water quality.

K_d values were also estimated for equilibrium metals partitioning between bed sediment and porewater for different solids/water ratios at key locations in the creek based on laboratory adsorption/desorption batch test results (CDM, 2001b). These values were within an order of magnitude and generally in agreement with the calibrated model values. USGS data from the synoptic survey during much higher flows (flows of approximately 3 m^3 s^{-1}) in June 1997 were used for model validation.

A good calibration was generally achieved for all of the metal concentrations. In almost all cases modeled concentrations of both total and dissolved metal were within 25% of values observed during June 2000, and in most cases were within 10%. A very good calibration was achieved for As, Cd (Figure 4 as an example), and Zn, but Pb and Cu results were not as close. The reasons for this are not clear. Correlation coefficients between modeled and measured values along the mainstem

TABLE III

Model efficiencies for metal concentrations ($\mu g\,L^{-1}$)

	As		Cd		Cu		Pb		Zn	
	Dissolved	Total	Dissolved	Total	Dissolved	Total	Dissolved	Total	Dissolved	Total
R^2 – Correlation Coefficient	0.90	0.87	0.98	0.99	0.72	0.90	0.94	0.93	0.91	0.90
Standard Error	4.04	6.38	0.19	0.17	0.98	0.70	0.76	0.98	92.79	99.86
Observed Mean	13.97	17.56	1.27	1.41	4.36	5.02	1.51	3.13	257.31	245.29
Standard Error % of Mean	29	36	15	12	22	14	51	31	36	41

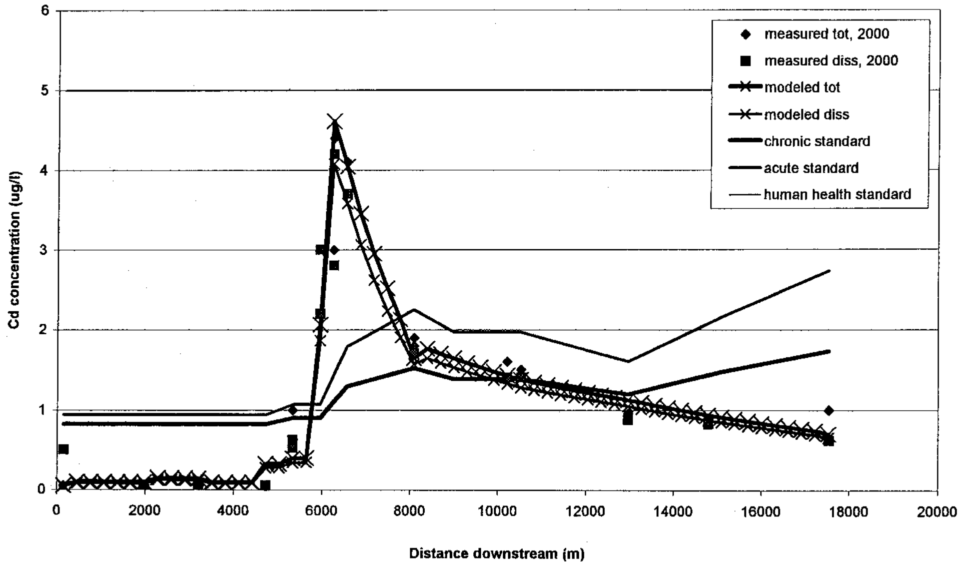

Figure 4. Comparison of measured (June 2000) and modeled Cd concentrations vs. distance in Tenmile Creek mainstem based on calibrated model, with applicable standards.

ranged from 0.72 for dissolved Cu (all others were greater than or equal to 0.87) to 0.99 for total Cd (Table III, Figure 5 for Cd). Standard errors ranged from 12% to 51% of measured mean concentrations for total Cd and dissolved Pb, respectively. The standard errors for dissolved and total Zn were relatively high (36 and 41%, respectively). However, most of these errors result from relatively large errors in the modeled peak values at one monitoring station approximately 6200 m downstream from the headwaters. The modeled peak dissolved and total Zn concentrations are displaced only 200 m upstream from the actual peaks at this location.

As part of validation, a close match was generally achieved for modeled and observed flows in the mainstem. Because flows were under-predicted by approximately 10% 5000 m downstream of the headwaters, additional water sources in this area were indicated that are not accounted for in the model. Flows were also over-predicted by approximately 8% 17 000 m downstream of the headwaters. This indicated some flow losses in this area of the mainstem. Good validation was achieved for most of the metals (see Figure 6 for Cd), but in the Rimini area, the peak total concentrations for Zn were significantly under-predicted, and for Cu were over-predicted. Uncertainties in some of the model inputs, including point source loading and tributary flow data, could be a factor in these results. Another important factor could be additional nonpoint source metals loadings associated with erosion and sediment transport during higher flows. These loadings were not explicitly modeled as part of the steady-state, low-flow study. The model is believed to be a very useful tool, however, for prediction of metals concentrations and loadings under low- or base-flow conditions. In addition, this validation in-

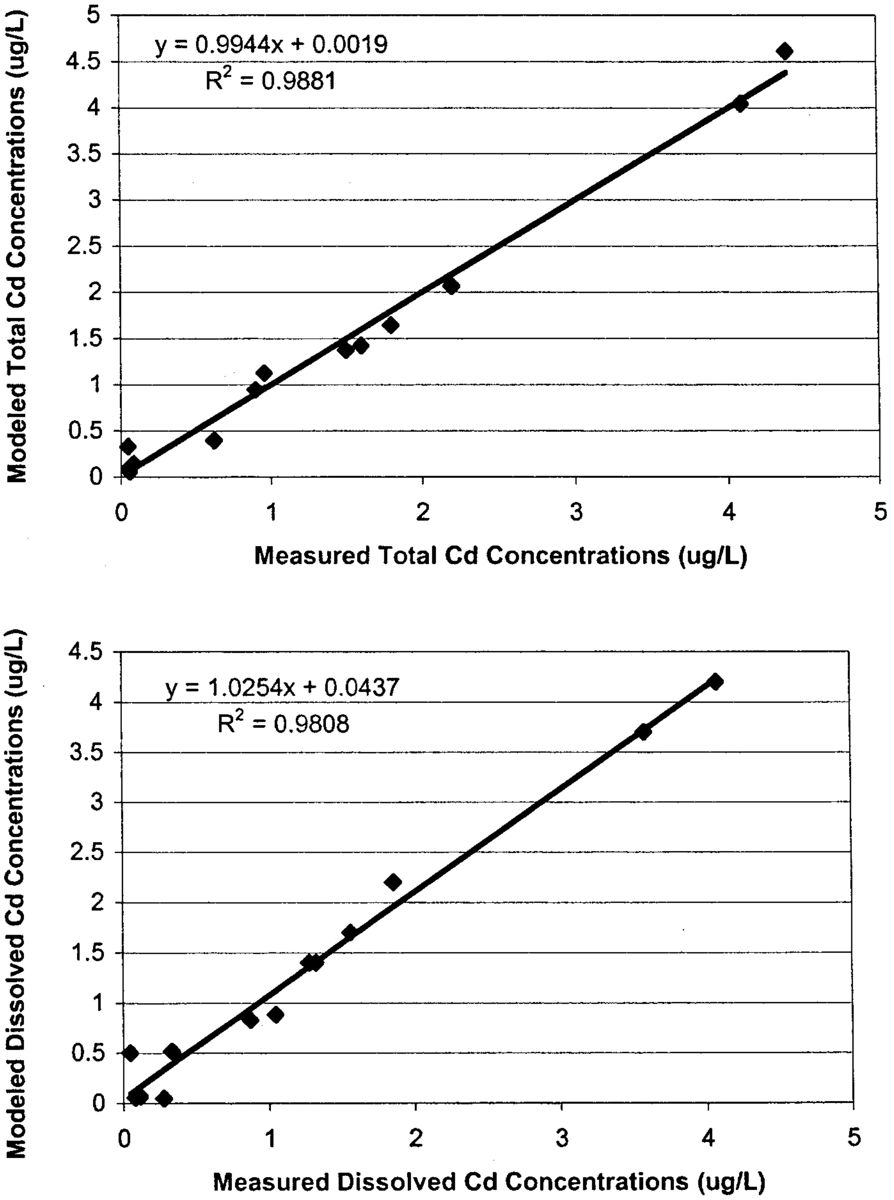

Figure 5. Measured vs. modeled concentrations for (a) total Cd and (b) dissolved Cd along the mainstem of Upper Tenmile Creek based on calibrated model.

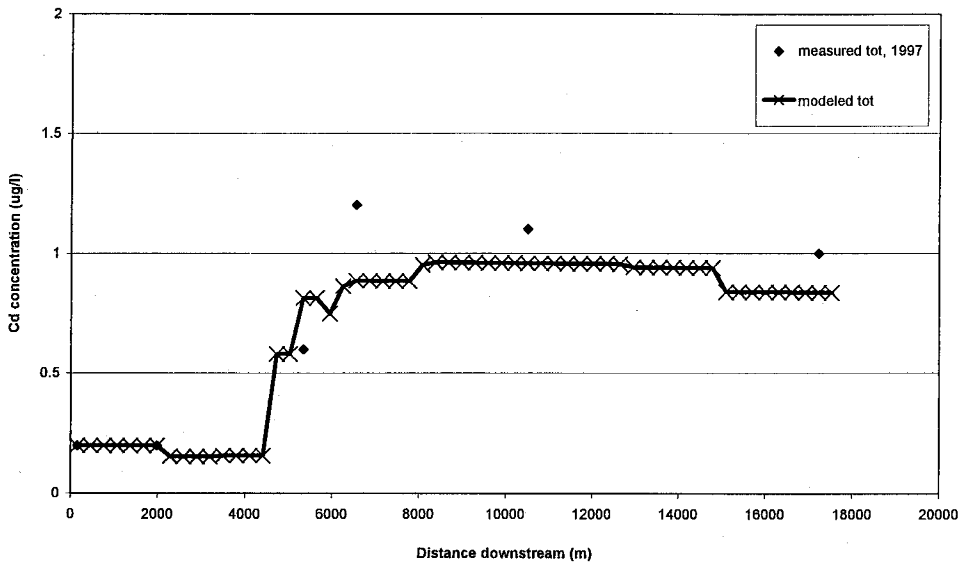

Figure 6. Comparison of measured (June 2000) and modeled Cd concentrations vs. distance in Tenmile Creek mainstem based on validated model.

formation plays a very important role in the identification and evaluation of these unaccounted nonpoint source metals and sediment loadings.

Monitoring, calibration, and validation results indicate that point sources such as adits, especially at Redwater Adit, Suzie Load, and near Lee Mountain, contribute high metals loadings to the mainstem both during baseflow and higher flows. In some locations, however, loadings can also be derived from shallow groundwater and bed sediment. Water quality criteria are exceeded at numerous locations throughout the stream. Adsorption and precipitation onto bed sediments of dissolved metals can also contribute to losses from the water column in some key areas. Although the details of monitoring results for other analytes, such as pH, oxidizing/reducing conditions, and electrical conductivity, are not presented here, some typical and interesting patterns and relationships with metals can be observed. High total metals loadings to the mainstem from point sources include primarily elevated dissolved metals associated with low pH and reducing conditions in the subsurface water. As this water enters the mainstem, total loadings and concentrations in the mainstem increase significantly. A large proportion of these metals come out of solution, however, through adsorption and/or precipitation reactions as this water mixes with mainstem water with greater pH and oxidizing conditions. Most of the total metals loadings and concentrations in the mainstem downstream from here are therefore comprised of adsorbed metals with dissolved metals contributing a smaller proportion of the total. The total metals concentrations also decrease with distance downstream as more metals are adsorbed or precipitated onto the bed sediments. There was some uncertainty in K_d values, the importance of precipitation reactions, and specific locations of unaccounted metals sources and losses associated with the development and application of the model.

Results of modeling the restoration alternatives generally showed that removal or reduction of point sources (such as adits) can significantly reduce most metals concentrations in the mainstem. For some metals (such as Zn), however, removal or stabilization of mine waste near watercourses and streambed sediment may also be needed. Results indicate that in order to meet all water quality criteria, particularly for Zn, modifications to the Helena water supply scheme and increasing baseflow in the mainstem will probably be necessary. Results of modeling the restoration alternatives are discussed below for each of the metals.

Arsenic. Modeling results show that all of the alternatives decrease total As concentrations relative to measured values along the mainstem downstream from the headwaters (Figure 7a). Alternative 6 was not modeled because Alternative 5 resulted in acceptable concentrations (below the human health standard of 18 μg L^{-1}) at all locations. Alternative 2 by itself has the least effect on concentrations, and values remain elevated above the standard in the Rimini subarea and further downstream. Alternative 5 results in the greatest decrease in concentrations (from >40 to <5 μg L^{-1} downstream from the Suzie Load). Values decrease below the standard along the entire length of the creek, although there is a spike in the concentration at approximately 8000 m downstream from the headwaters that almost exceeds the standard. Alternatives 1 and 4 also reduce concentrations below the standard along the entire length except at this location, where values can increase to >30 μg L^{-1}.

Cadmium. All of the alternatives decrease total Cd concentrations relative to measured values (Figure 7b). Alternatives 4 and 5 have the greatest effect on decreasing values. Concentrations decrease below the aquatic life chronic standard, which varies along the length of the creek with hardness, with distance along the creek. Alternatives 1, 2, and 3 also reduce concentrations significantly, but values (up to >2 μg L^{-1}) remain above the standard in the Rimini subarea.

Copper. All alternatives reduce total Cu concentrations considerably, with Alternatives 5 and 6 generally reducing values the most (Figure 7c). All alternatives, however, show total Cu concentrations (up to approximately 6 μg L^{-1}) elevated above the aquatic life chronic and/or acute standard near the Redwater Adit and Suzie Load. Concentrations vary, depending on the alternative, further than 10 000 m downstream from the headwaters. However, all are below the chronic standard (<4.5 μg L^{-1}) further than approximately 13 000 m downstream from the headwaters.

Lead. Alternative 6 has the greatest effect on reducing total Pb concentrations (Figure 7d). However, values (up to approximately 2 μg L^{-1}) remain above the aquatic life chronic standard along most of the length of the creek. Total Pb concentrations (up to about 4 μg L^{-1}) are considerably above the criterion in the Rimini area for alternatives 1, 2, 4, and 5.

Zinc. All alternatives reduce total Zn concentrations considerably (Figure 7e). Al-

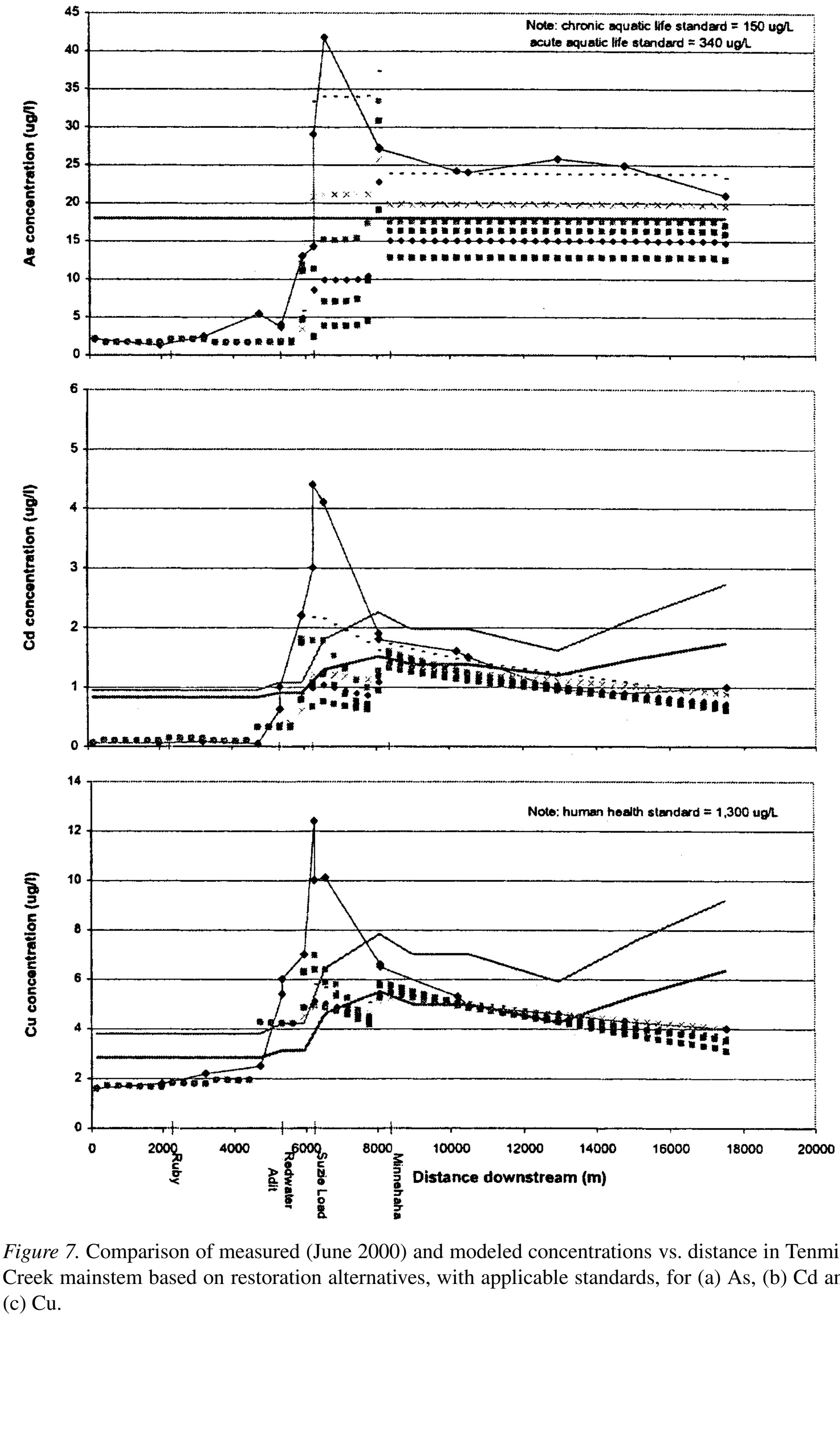

Figure 7. Comparison of measured (June 2000) and modeled concentrations vs. distance in Tenmile Creek mainstem based on restoration alternatives, with applicable standards, for (a) As, (b) Cd and (c) Cu.

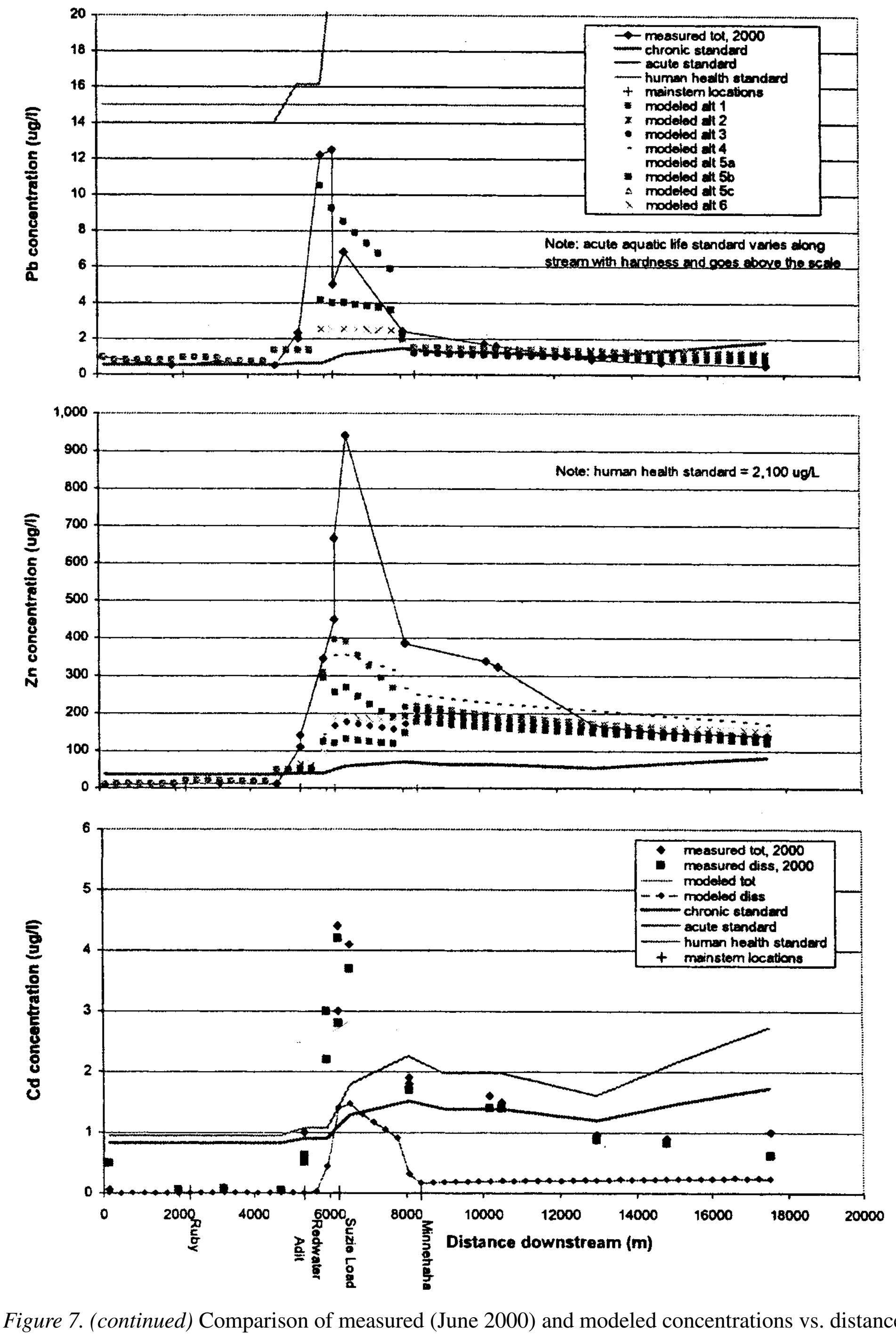

Figure 7. (continued) Comparison of measured (June 2000) and modeled concentrations vs. distance in Tenmile Creek mainstem based on restoration alternatives, with applicable standards, for (d) Pb, (e) Zn, and (f) Cd based only on bed sediment contamination.

ternative 6 has the greatest effect on reducing concentrations, but all values (up to about 350 μg L^{-1}) in the Rimini subarea and farther downstream remain above the aquatic life acute and chronic criteria (these two standards are equal for Zn).

Results of modeling the effects of bed sediment metal concentrations on the Tenmile Creek mainstem generally indicate that contaminated sediment contributes a relatively small percentage of the total metals loading to the water column under baseflow conditions (see Figure 7f for Cd). The exception appears to be As. A large amount of As has the potential to desorb from the sediments and cause dissolved concentrations (up to 80 μg L^{-1}) to exceed the human health standard of 18 μg L^{-1}, as well as to exceed existing concentrations (up to about 25 μg L^{-1}). However, the reason that the modeled results exceed observed values is not clear. Total and dissolved Cd and Pb concentrations (up to approximately 1.5 and 3 μg L^{-1}, respectively) can also exceed the chronic standards in the vicinity of the Suzie Load, and Zn approaches the chronic and acute standards. Cu remains below all standards.

Results of estimating equilibrium metals concentrations in the water column at the four selected locations based on the equilibrium porewater concentrations from the desorption batch tests indicate that exceedances of the standards could occur for these constituents for most of the locations evaluated (Table IV). This is based only on desorption of particulate metals from the bed sediments into the water column, under steady-state, low flow conditions. These values could most likely exist in near-stagnant water under extreme low-flow conditions. These results are in general agreement with the modeling results discussed above, but in most cases are higher. This difference is probably due to the fact that the model simulates advective flow that reduces water column concentrations, whereas the second equilibrium method does not take into account flowing water, resulting in higher, more conservative concentration estimates.

Results of the probabilistic approach to evaluate the time required to scour and flush contaminated bed sediments out of Tenmile Creek past Rimini based on flood events indicate that a 1-yr flood (MAF) could flush approximately 884 tonnes of contaminated sediment (assuming contaminated sediment is 1.61 km (1 mile) long, 0.3 m (1-ft) thick, and 1.22 m (4-ft wide); Table V). It would take approximately 1.5 days of a 1-yr flood (assuming that the peak flow of a 1 yr flood is constant over 1.5 days) to flush all of the sediment. If the peak flow occurs over a 24-hr period, it would take, on average, 1.5 one-year floods to remove this sediment. Therefore, there is approximately a 99% probability of flushing 65% of the sediment in any given year (assuming no additional inputs of contaminated sediment). A 5-yr flood could flush approximately 1007 tonnes/per day of sediment, and it would take approximately 1.35 days of a 5-yr flood (assuming the peak flow is constant over 1.35 days) to flush all of the sediment. If the peak flow occurs over a 24-hr period, it would take, on average, 1.35 five-year floods to remove this sediment. There is approximately a 20% probability of flushing 73% of the sediment in any given year. Lastly, a 10-yr flood could flush approximately 1291 tonnes/per day of sediment.

TABLE IV

Initial benthic porewater concentrations, equibrium metal concentrations in Upper Tenmile Creek water from bed sediment metals, and standards (all in μ L^{-1})

Segment	Pore water concentrations						Water concentrations					
	Al	As	Cd	Cu	Pb	Zn	Al	As	Cd	Cu	Pb	Zn
Segment 61	2100	600	7	35	140	700	420	**120**	**1.4**	**7**	**28**	**140**
Segment 62	900	120	5	30	50	800	180	**24**	1	**6**	**10**	**160**
Tenmile below Suzie Adit	630	200	2	11	55	100	126	**40**	0.4	2.2	**11**	20
Poison Creek near Red Mountain	830	205	14	90	160	1450	166	**41**	**2.8**	**18**	**32**	**290**
Standard	NS	NS	NS	NS	NS	NS	NS	18	1.14	4.03	0.94	52.17

Pore water concentrations from batch (desorption) tests.
Based on sediment depth of 0.3 m (1 ft), porosity of 0.25, and water depth of 0.3 m (1 ft).
Values in bold exceed standards.
All standards are aquatic life chronic criteria based on average hardness in Tenmile Creek, except As, which is human health standard.
NS = No standard.

TABLE V
Estimation of scour and flushing of contaminated sediment from Tenmile Creek

Flood return period (yr)	Q (cfs)	V (ft s^{-1})	S (t day^{-1})	% Removal for 1 flood	% Probability of occurring in any given year	# of floods required
1	286	8	884	65	>95	1.5
5	364	8.3	1007	73	20	1.35
10	511	9	1291	100	10	1

Assumptions:
Trapezoidal channel with 1:1 side slopes and 1.22 m (4 ft) bottom width.
Stream slope (S) = 0.0238.
Manning's n/Roughness coefficient (n) = 0.05.
Contaminated bed sediment depth = 0.3 m (1 ft), width = 1.22 m (4 ft), and length = 1.61 km (1 mile) (total volume of 595 m^3).
Contaminated bed sediment is predominantly fine-grained, i.e., sand-size particles (methods are currently not available for evaluation of discharge of silt and clay-size particles).
Bed sediment density = 2.5 kg L^{-1}.

It would take approximately 1 day of a 10-yr flood (assuming the peak flow is constant over 1 day) to flush all of the sediment, so there is approximately a 10% probability of flushing all of the sediment in any given year. In reality, these peak flood flows would not be constant over a 1-day period, so it would probably take longer than these estimates.

5. Conclusions

This study has demonstrated the methods and results for simulating metals fate and transport and the effectiveness of potential restoration alternatives in a mining-impacted mountain watershed, using the U.S. EPA WASP5 model and the Upper Tenmile Creek, Montana as a case study. This model has proven to be critical for planning effective and sustainable restoration measures in this watershed. The process of model development, calibration, validation, and application discussed is useful for understanding the Tenmile Creek system as well as quantitative evaluation of restoration alternatives. The WASP5 model and methodology used in this study can be applied to other watersheds where metals and other toxic contaminants are problems to aid in evaluating contaminant sources, transport, impacts, uncertainties, and restoration measures.

The model generally confirmed observed metals concentrations in the mainstem for both the June 2000 and June 1997 sampling events. Flows and metals loadings that are unaccounted for, as well as uncertainties in the watershed, sampling results, and in the model itself, were identified and evaluated using the model. Many of the important processes in the creek and watershed were also identified. Lastly, the potential effectiveness of restoration alternatives have been quantified and evaluated using the model.

This model of the Upper Tenmile Creek system will probably be revised as necessary in the future to improve the accuracy of results for planning and design of restoration measures, as well as for application to other management issues, such as TMDL development. It will also be used to evaluate sediment and associated metals loadings under high-flow conditions during snowmelt and storm runoff. This can be accomplished by developing and implementing a dynamic version of the model and using water and sediment loads under high-flow conditions as new input parameters. The water and sediment loads, however, must currently be estimated outside of the WASP model using separate models or other estimation methods, such as the Revised Universal Soil Loss Equation (RUSLE) for erosion and the Rational Method for peak flows. Additional sampling could be performed to evaluate in more detail flows, loadings, and losses that are currently unaccounted for in the model. This should include sampling high-flow events to calibrate a dynamic model. The new WASP module Metal Exposure and Transformation Assessment (META4; Martin and Medine, in preparation) could also be incorporated into the model for future application. In addition to adsorption/desorption, META4 explicitly models precipitation/dissolution reactions that can be very important in metals fate and transport in mined watersheds, particularly those with extreme and highly variable pH conditions.

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